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How to include the impact of climate change in the extinction risk assessment of policy plant species?

Fabio Attorre[a], Thomas Abeli[b], Gianluigi Bacchetta[c], Alessio Farcomeni[d], Giuseppe Fenu[c], Michele De Sanctis[a], Domenico Gargano[e,*], Lorenzo Peruzzi[f], Chiara Montagnani[g], Graziano Rossi[b], Fabio Conti[h], Simone Orsenigo[i]

[a] Department of Environmental Biology, Sapienza University of Rome, P.le A. Moro 5, 00185, Roma, Italy
[b] Department of Earth and Environmental Sciences, University of Pavia, Via S. Epifanio 14, 27100, Pavia, Italy
[c] Centre for the Conservation of Biodiversity (CCB), Department of Life and Environmental Sciences, University of Cagliari, Italy
[d] Department of Public Health and Infectious Diseases, Sapienza University of Rome, Rome, Italy
[e] Department of Biology, Ecology and Earth Sciences, University of Calabria, Arcavacata di Rende, Cosenza, Italy
[f] Department of Biology, University of Pisa, Pisa, Italy
[g] Department of Earth and Environmental Sciences University of Milano-Bicocca, Milan, Italy
[h] School of Biosciences and Veterinary Medicine, University of Camerino – Floristic Research Center of the Apennines, National Park of Gran Sasso and Laga mountains, San Colombo, Barisciano, L'Aquila, Italy
[i] Agricultural and Environmental Sciences – Production, Landscape, Agroenergy, University of Milan, Milan, Italy

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ABSTRACT

Climate change can have significant impacts on the survival of plant species. However, it is seldom included in the assessment of the extinction risk according to IUCN Red List criteria. Lack of data and uncertainties of predictions make difficult such inclusion. In our paper we present an approach, in which the effect of climate change on plant species spatial distribution is used to prioritize conservation within IUCN categories. We used, as a case study, 37 Italian policy species, relevant for conservation, and listed in the Habitat Directive and Bern Convention, and for which a Red List (RL) assessment was available. A stochastic SDM incorporating data on plant dispersal, generation length, and habitat fragmentation was used to predict a range shift due to climate change according to two climatic scenarios (RCP 2.6 and 8.5). No species was predicted to become extinct in the considered timespans (2050 and 2070) due to climate change, and only two were characterized by critical decline probabilities. However, all taxa were potentially affected by climate change through a reduction of their range. In all RL categories, species with the highest predicted reduction of range were those from lowlands, where fragmentation of natural habitats has occurred in the last decades. In these cases, despite some limitations, assisted migration can be considered a suitable conservation option.

1. Introduction

The European Union has one of the most advanced and effective intergovernmental biodiversity policies (Beresford, Buchanan, Sanderson, Jefferson, & Donald, 2016). The "Habitat" Directive 92/43/CEE (hereafter HD) represents the core strategy of nature conservation in Europe, aiming at protecting, maintaining or restoring a "favourable" conservation status for policy species (*taxa* of flora and fauna included in the Habitat Directive 92/43/EEC and the Bern Convention annexes). However, previous reports at national and European levels demonstrated that several policy species meet an "unfavourable" conservation status (Condé, Jones-Walters, Torre-Marin, & Romao, 2010; Fenu et al., 2017). This is supported by different red lists at national (Moreno Saiz, Domìnguez Lozano, & Sainz Ollero, 2003; Rossi et al., 2016) and EU levels (Bilz, Kell, Maxted, & Lansdown, 2011; García Criado et al., 2017). Among threats affecting the policy species, climate change currently causes minor effect (Bilz et al., 2011; Fenu et al., 2017; Rossi et al., 2016; Thuiller, 2007). Nonetheless, global warming is increasing its negative impacts, with new temperature records set every year (e.g., CNR's annual climatic reports for Italy http://www.isac.cnr.it/climstor/climate/; NOAA's Global Climate Report https://www.ncdc.noaa.gov/sotc/global/; Feng et al., 2014; Pauli et al., 2012).

* Corresponding author. Current address: Museo di Storia Naturale della Calabria ed Orto Botanico dell'Università della Calabria, loc. Polifunzionale, I-87036, Arcavacata di Rende, Italy.
E-mail address: domenico.gargano@unical.it (D. Gargano).

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Table 1
List of the *Policy Species* and parameters considered in the analyses. Dispersal mechanisms: boleochory (diaspores are thrown away from moving capsules), meteorochory (diaspores are dispersed by wind), epichory (diaspores are dispersed by animal), myrmecochory (diaspores are dispersed by ants). D: mean and maximum dispersal distance in meters. Maturity: number of years required by the species to reach the sexual maturity. Kernel: dispersal Kernel function based on the species dispersal syndrome. No. occ.: number of georeferenced occurrences used in the model. Mean alt.: mean altitude of the occurrence sites.

Species	Family	Dispersal mechanism	Source	D Mean (m)	D Max (m)	Kernel	Maturity (years)	No. occ.	Mean Alt. (m a.s.l.)
Adenophora liliifolia (L.) Ledeb. ex A.DC.	Campanulaceae	Boleochory	Landolt et al. (2010)	1	5	NA	3	87	819.1
Aquilegia alpina L.	Ranunculaceae	Boleochory	Landolt et al. (2010)	1	5	NA	4	142	1884.4
Arnica montana L. subsp. *montana*	Asteraceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	2	1318	1749.4
Artemisia genipi Weber ex Stechm.	Asteraceae	Boleochory /Epichory	Landolt et al. (2010)	1	5	NA	3	227	2418.4
Asplenium adulterinum Milde subsp. *adulterinum*	Aspleniaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	3	65	1307.3
Brassica glabrescens Poldini	Brassicaceae	Boleochory	Landolt et al. (2010)	1	5	NA	2	30	158.1
Brassica insularis Moris	Brassicaceae	Boleochory	Santo, Fenu, Domina, and Bacchetta (2013)	1	5	NA	4	33	273.2
Campanula morettiana Rchb.	Campanulaceae	Boleochory	Landolt et al. (2010)	1	5	NA	3	134	1771.3
Campanula sabatia De Not.	Campanulaceae	Boleochory	Landolt et al. (2010)	1	5	NA	3	48	554.2
Carex panormitana Guss.	Cyperaceae	Meteorochory	Urbani, Calvia, and Pisanu (2013)	40	150	Weibull	5	31	447.0
Crocus etruscus Parl.	Iridaceae	Barochory/ Myrmecochory	Carta, Moretti, Nardi, Siljak-Yakovlev, and Peruzzi (2015)	1	5	NA	5	38	327.1
Cypripedium calceolus L.	Orchidaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	15	202	1502.6
Daphne petraea Leyb.	Thymelaeaceae	Myrmecochory	Landolt et al. (2010)	1	5	NA	10	40	1143.9
Dianthus rupicola Biv. subsp. *rupicola*	Caryophyllaceae	Boleochory	Expert observation	1	5	NA	4	53	125.0
Diphasiastrum alpinum (L.) Holub	Lycopodiaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	10	37	2104.2
Euphrasia marchesettii Wettst. exMarches.	Orobanchaceae	Boleochory	Landolt et al. (2010)	1	5	NA	1	53	66.5
Fritillaria montana Hoppe ex W.D.J.Koch	Liliaceae	Boleochory	Landolt et al. (2010)	1	5	NA	5	92	1019.3
Gentiana ligustica R.Vilm. & Chopinet	Gentianaceae	Boleochory	Landolt et al. (2010)	1	5	NA	3	66	1349.4
Gentiana lutea L. subsp. *lutea*	Gentianaceae	Boleochory /Meteorochory	Landolt et al. (2010)	1	5	NA	5	452	1598.4
Gentiana lutea L. subsp. *vardjanii* Wraber	Gentianaceae	Boleochory /Meteorochory	Landolt et al. (2010)	1	5	NA	5	77	1479.2
Gladiolus palustris Gaudin	Iridaceae	Boleochory	Landolt et al. (2010)	1	5	NA	3	207	489.1
Huperzia selago (L.) Bernh. ex Schrank & Mart. subsp. *selago*	Lycopodiaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	10	474	1441.9
Linaria flava (Poir.) Desf. subsp. *sardoa* (Sommier) A.Terracc.	Plantaginaceae	Unknown	Pinna et al. (2012)	1	5	NA	1	30	58.7
Lycopodium annotinum L. subsp. *annotinum*	Lycopodiaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	10	1073	1650.2
Lycopodium clavatum L.	Lycopodiaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	10	349	1617.4
Ophrys lunulata Parl.	Orchidaceae	Meteorochory	Unpublished data	10	500	Gaussian	15	51	430.5
Physoplexis comosa (L.) Schur	Campanulaceae	Boleochory	Landolt et al. (2010)	1	5	NA	3	572	1494.1
Pilularia minuta Durieu ex A.Braun	Marsileaceae	Unknown	Expert observation	1	5	NA	3	32	407.7
Primula polliniana Moretti (= *P. spectabilis* Tratt.)	Primulaceae	Unknown	Landolt et al. (2010)	1	5	NA	3	132	1532.3
Saxifraga florulenta Moretti	Saxifragaceae	Boleochory	Landolt et al. (2010)	1	5	NA	20	42	2322.0
Saxifraga presolanensis Engl.	Saxifragaceae	Boleochory	Landolt et al. (2010)	1	5	NA	5	38	1918.0
Saxifraga tombeanensis Boiss. ex Engl.	Saxifragaceae	Boleochory	Landolt et al. (2010)	1	5	NA	5	30	1394.8
Selaginella denticulata (L.) Spring	Selaginellaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	3	104	404.4
Selaginella helvetica (L.) Spring	Selaginellaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	3	49	870.8
Selaginella selaginoides (L.) P.Beauv. ex Schrank & Mart.	Selaginellaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	3	43	1313.4
Spiranthes aestivalis (Poir.) Rich.	Orchidaceae	Meteorochory	Landolt et al. (2010)	10	500	Gaussian	15	106	309.1
Stipa austroitalica Martinovský	Poaceae	Meteorochory	Expert observation	40	150	Weibull	2	50	462.0

The ways climate change affects plants are various and strongly interact with other factors such as species traits, human disturbance, including habitat fragmentation, magnitude of extreme events, etc. (e.g. Honnay et al., 2002; Niu et al., 2014; Orsenigo, Mondoni, Rossi, & Abeli, 2014). Thus, understanding how climate change will affect policy species is of primary importance to define current conservation policy and effective actions plans, to avoid species extinction, to identify future mismatch between protected areas and species distribution (Araùjo, Alagador, Cabeza, Nogues-Bravo, & Thuiller, 2011; Fois, Bacchetta, Cogoni, & Fenu, 2018), and to mitigate the impact of fragmentation on species range shift.

The effects of climate change on future plant distributions is typically assessed by applying species distribution models (SDMs hereafter) to projected climatic conditions (e.g., Attorre et al., 2011; Benito Garzón, Sánchez de Dios, & Sáinz-Ollero, 2008; Ferrarini, Rossi, Mondoni, & Orsenigo, 2014; Fois et al., 2018). Based on SDMs, several approaches for predicting climate change impacts on species extinction risk, according to the IUCN categories and criteria have been applied (see Draper Munt, Muñoz-Rodríguez, Marques, & Moreno Saiz, 2016; Fois et al., 2018; Thuiller, Lavorel, Araújo, Sykes, & Prentice, 2005). This approach has been questioned by Akçakaya, Butchart, Mace, Stuart, and Hilton-Taylor (2006) because of the misuse of IUCN criteria.

Despite this, some authors recently used coupled niche-demographic models to assess the impact of climate change on the extinction risk of a number of reptile and amphibian species according to IUCN categories and criteria (Keith et al., 2014; Stanton, Shoemaker, Pearson, & Akçakaya, 2015). However, applicability of SDMs to red lists is difficult due to model uncertainties, as many biotic and abiotic factors cannot (or are difficult to) be included in these models such as, for instance, soil features, competition and mutualism (i.e. Fordham et al., 2012), as well as genetic adaptation of target species. For this reason, we suggest that the two methods can complement each other: red list categories provide information on both the current and future extinction risk for a target species, while projected SDMs may provide warnings on the magnitude of future extinction risk. For instance, under climate change, species belonging to the same red list category may show different future projected range loss or gain, implying a different urgency in the application of conservation measures. This should lead to more accurate conservation prioritization of group of species.

In our paper, we used such an approach on Italian policy plant species for which a comprehensive assessment of the conservation status has been recently conducted based on IUCN categories and criteria (Rossi et al., 2016). Our approach was based on the application of a cellular automaton model to produce a stochastic distribution of potential dispersal outcomes of range shift of such species in Italy. This model can produce quite realistic dispersal patterns in predicting future plant species distributions, by incorporating movement into SDMs using species-specific demographic information and dispersal limitation as well as habitat heterogeneity (see also Miller, Holloway, & Gillins, 2015 for theoretical background, and Benito, Lorite, Pérez-Pérez, Gómez-Aparicio, & Peñas, 2014; Di Traglia, Attorre, Francesconi, Valenti, & Vitale, 2011; Engler et al., 2009; Summers, Bryan, Crossman, & Mayer, 2012 for practical examples).

2. Materials and methods

2.1. Data set

A list of 107 policy vascular plants occurring in Italy was considered. The taxonomic treatment of species and subspecies follows Bartolucci et al. (2018). Taxa occurrence data were obtained from the Italian Red List geodatabase (Rossi et al., 2016), and from a network of databases with georeferenced occurrences (Agrillo et al., 2017; Bedini et al., 2016; Martellos et al., 2011). Taxa with fewer than 30 occurrences were excluded due to the high potential inaccuracy of the model. This significantly limits the applicability of the approach to rare species with few locations, even though in the red listing process such species are more likely to be included in threat categories. Moreover, a further four very widespread species, namely *Galanthus nivalis* L., *Himantoglossum adriaticum* H. Baumann, *Orchis provincialis* Balb. ex Lam. & DC. and *Ruscus aculeatus* L. were excluded because the available information was likely to be not representative of their current Italian distribution and ecological niche. Overall, we retained 37 taxa for which a final data set of 6631 occurrences, ranging from 30 to 1318 per species, was obtained (Table 1). Based on Vittoz and Engler (2007), we assigned a mean and a maximum dispersal distance for each plant species. Dispersal kernel were based on dispersal syndrome (Table 1), according to Flora Indicativa (Landolt et al., 2010) and specific literature on the biology of each considered taxon (Table 1). In cases where data about dispersal distance was missing in the literature, we assigned it based on expert observation. Finally, we included in the model the estimated time (from 1 to 20 years) needed by each species to reach sexual maturity.

2.2. Environmental variables

Current (1960–1990, Hijmans, Cameron, Parra, Jones, & Jarvis, 2005) and future climate-change simulations produced by HadGEM2-AO Global Circulation Models were obtained at the maximum resolution available, namely 30 arc second ($\sim 1\ km^2$) from Worldclim (http://www.worldclim.org/). Two representative concentration pathways (RCP), specifically the lowest emission scenario RCP2.6, and the highest emission scenario RCP8.5, were considered to include the variability of climatic predictions, and two-time periods 2050 (average for 2041–2060) and 2070 (average for 2061–2080).

Climatic variables included were annual, spring, summer, autumn and winter precipitation and mean, minimum and maximum temperatures. Slope and geological maps were included as further environmental variables. Geological maps were considered as a surrogate of pedological information that was not available for the overall study area. The twenty-three lithological typologies, such as they appear in the Italian lithological map 1:250.000, have been clustered in five classes: Clastic formations; Carbonatic; Volcanic; Arenaceous; and, Clayey. This aggregation was carried out to simplify the highly differentiated Italian lithology (Attorre et al., 2014). Landscape configuration was computed by reclassifying the categories of the Corine Land Cover map for 2006 into two classes, including natural and semi-natural (i.e. forests, shrublands, grasslands) and artificial (i.e. agricultural land and urban areas) landscapes.

2.3. Data analysis

In our study, Random Forest (RF, Breiman, 2001) was used as SDM. In many comparative analyses RF has proved to be efficient in predicting the spatial distribution of plant species (e.g., Attorre et al., 2013; Barbet-Massin, Jiguet, Albert, & Thuiller, 2012; Williams et al., 2009) because of its features such as bootstrap-resampling, tree averaging and randomisation of predictors. Moreover, since the presence-only models and models without dispersal information tend to over predict the spatial distribution of species (Vaclavik & Meentemeyer, 2009), we used a model based on presence/absence data, that has already demonstrated its efficacy in modeling the distribution of rare and threatened plant species (Fois et al., 2017; Scarnati, Attorre, Farcomeni, Francesconi, & De Sanctis, 2009). We assessed the ability of RF to discriminate between presences and absences via area under the curve (AUC) using k-fold cross-validation (five groups). SDMs were projected over the future climatic variables for both climate change scenarios to represent future habitat-suitability change.

2.4. Simulating dispersal, colonization and local extinction

The simulations used a cell by cell sampling. For each iteration and for each cell we assessed the presence/absence of each species. If the species was present at the previous time-occasion, the dispersal kernel of the species was used by generating a random number from the dispersal kernel (e.g., if the kernel was Gaussian with a certain mean and variance, a species-cell specific dispersal was generated from a Gaussian with those parameters). For all cells within the current randomly generated dispersal distance we computed a probability of presence. This was the product of the habitat-suitability (which could be zero) and the predicted presence probability. Then, for each cell within the dispersal distance and the current cell the presence at the current time-occasion was generated as a Bernoulli trial with the cell-specific probability of presence, independently of other cells. If the Bernoulli trial randomly generated an absence for the current cell, the cell was marked as locally extinct for the simulation replica under consideration.

2.5. Cellular automation and simulation steps

The simulation scheme described above was replicated 10,000 times to explore different dispersal scenarios and consider variability due to randomness. The probabilities reported were finally estimated as averages over the 10,000 replicates.

2.6. Sensitivity analysis

Cellular automation was repeated with different scenarios to evaluate the effects of the parameters assumed for the simulation. We varied parameters for the dispersal kernels, randomly perturbed suitability values, and finally evaluated the variability due to these exogenous parameters.

2.7. Evaluation of potential extinction risk

For each realization, the number of grid cell lost, potentially gained by each taxon or those where it was proved to be stable in different climate-change scenarios was summed. A taxon was considered 'extinct' when it reached 0% of initial presence, while a 'critical range decline' was attributed to taxa whose potential range was reduced to 10% (Engler et al., 2009). The probability of persistence, critical range decline and extinction was computed for each taxon and for the two-time slices (2050 and 2070) by averaging all realizations. The influence of the altitudinal range and dispersal mechanism on range shift predictions was also estimated.

3. Results

The majority of selected taxa is characteristic of alpine and subalpine environments, and generally shows a boleochorous dispersal syndrome (Table 1). RF showed a very high discrimination ability with an out-of-bag cross-validated AUC median for the 37 species of 0.9988 (max 1, min 0.971). The average probability of extinction and critical range decline (future projected range reaching 10% of the original range) is reported in Table 2. No species was predicted to become extinct in the considered timespan due to climate change, and only *Crocus etruscus* and *Stipa austroitalica* were characterized by critical decline probabilities (Table 2). However, similarly for the two climatic scenarios, all taxa were potentially affected by climate change through a reduction of their range, from 84% for *Stipa austroitalica* to 12% for *Brassica insularis*, with an average loss across the species of 48.5%. No significant difference in range loss was found according to the altitudinal range, while for RCP8.5 scenario only, the dispersal type showed a significant effect on species range decline, with boleochorous species being characterized by an area loss of 38% more ($p = 0.029$) when compared to the meteorochorous ones.

4. Discussion

Despite the increasing evidence of the impact of climate change on the survival of many species, such a threat is quantitatively considered in the red list assessment for only a small number of these species (Akçakaya, Butchart, Watson, & Pearson, 2014).

A possible explanation may be provided by the uncertainties in predicting the extinction risk using SDMs and climatic scenarios. By coupling population and distribution models, Stanton et al. (2015) used RL criteria to identify species at risk of extinction with decades of warning time. Notwithstanding such improvements, lack of data and modeling uncertainties still seem to constrain a routine use of RL system coupled with SDMs. According to our approach, predictions on range shift of species can complement the RL assessment to prioritize conservation actions within each RL category. Predictions for selected policy species have been produced by a stochastic dynamic species distribution model, which incorporated dispersal kernels and habitat fragmentation. Results indicate that no species is at serious risk of extinction over the time frame investigated here. Under the worst scenario, a significant range decline is predicted only for two taxa (*Crocus etruscus* and *Stipa austroitalica*, both Italian endemics, see Brundu et al., 2017; Harpke et al., 2015; Peruzzi, Conti, & Bartolucci, 2014; Peruzzi et al., 2015). Our study suggests *S. austroitalica* as an especially urgent conservation target, in order to reduce the foreseen drastic loss of range predicted in the next decades. Nevertheless, the range will decrease in all plant species, in some cases at an alarming rate, with average losses higher than 70% (i.e. besides the two species mentioned above: *Campanula morettiana, Linaria flava* subsp. *sardoa*, *Selaginella denticulata, Spiranthes aestivalis*; Table 2). The trends obtained for *Linaria flava* subsp. *sardoa* are similar with those obtained in a previous work realized in Sardinia using Maxent, which however showed a smaller reduction of future suitable areas for this plant (37.2%; Fois et al., 2018).

Overall, our results differ from other similar studies reporting an higher percentage of extinction risk (see Benito et al., 2014; Engler et al., 2009), but it can be explained considering the differences in the number of species analyzed, the spatial resolution used, the study area and the modeling approach. Indeed, our results are consistent with more recent models incorporating dispersal strategies, evolutionary potential, high-resolution spatial data, etc. and predicting long-term species survival in loco or through migration (e.g. Cotto et al., 2017; Hannah et al., 2014; Randin et al., 2009). According to Benito et al. (2014), dispersal mechanism was relevant in determining the decline of species with poor dispersal capacity (boleochorous), these being characterized by a greater area loss with respect to (meteorochorous) species with a greater capacity of colonizing suitable habitats (see also Engler et al., 2009).

When looking at the RL categories some interesting results emerged (Table 2). In fact, in all categories species with the greatest reduction of range grow in lowlands, namely *Spiranthes aestivalis, Pilularia minuta, Crocus etruscus* and *Stipa austroitalica*. This can be explained considering that these areas have been highly impacted during the last decades by human settlements and infrastructures, and the expansion of agricultural areas (Falcucci, Maiorano, & Boitani, 2007); here, the dispersal limitations found in many Mediterranean species can reduce their chance to colonize new suitable sites outside their current range. A similar result was obtained in a recent study carried out in Sardinia: the probability of extinction was higher along the coast and in plains areas; these areas were characterized by recent land use change from semi-natural into urbanized landscapes as a consequence of industrial settlements and increasing tourism development (Fois et al., 2017).

An example is provided by the relatively high extinction likelihood found for *Stipa austroitalica*. This species typically occurs in complex landscape mosaics at low and middle elevation. In such contexts, recent diachronic analyses (Gargano, Mingozzi, Massolo, Rinaldo, & Bernardo, 2012) revealed that land-use change patterns (i.e. increase of urban and agricultural areas, and expansion of shrubby vegetation) are causing a decline of the pastures with *Stipa austroitalica*. Then, the decline of this taxon is mainly related to abandonment of traditional agricultural practices like nomadic grazing; this decline could be significantly increased with the combined effect of climate change. Similarly, increasing forest surface in mountain regions promoted by land abandonment, and further favored by warmer temperatures (Lazpita & Gibon, 2000; Pearson & Dawson, 2003), could explain the range decline of species growing in open-habitats. *Crocus etruscus* shows large seeds, dispersed over short distances by ants (Carta, Moretti et al., 2015). The seeds require cool temperatures to grow after warm stratification (Carta, Probert, Moretti, Peruzzi, & Bedini, 2014). In addition, this species is mostly outcrossing (Carta, Flamini, Cioni, Pistelli, & Peruzzi, 2015) and self-incompatible (Carta, Campigli, Peruzzi, & Bedini, 2016), so that its survival maybe even further threatened by all these intrinsic biological features, after the expected significant range reduction due to climate change.

Species with the smallest predicted range reduction mainly grow on steep rocky habitats, *Saxifraga tombeanensis, Saxifraga florulenta, Primula spectabilis*, and *Dianthus rupicola*. The greater capacity of persistence of chasmophytes, that is plant species growing in the crevices of rocks, in a changing climate deserves further investigation, even though it might depend on the fact that these habitats are rather conservative and less subjected to vegetation dynamics (Pignatti & Pignatti, 2013).

Table 2
Outcomes of the analyses. Species are reported according to RL categories based on the assessment made by Rossi et al. (2016) and ordered according to the mean predicted range loss. Probability of extinction (Ex), of critical range decline (Cr) and of presence (Pr) within 2050 (50) and 2070 (70), according to two different representative concentration pathways (RCP).

	RCP2.6						RCP8.5						Range loss and current red list category	
Species	Ex 50	Ex 70	Cr 50	Cr 70	Pr 50	Pr 70	Ex 50	Ex 70	Cr 50	Cr 70	Pr 50	Pr 70	Mean loss	RL Cat
Spiranthes aestivalis (Poir.) Rich.	0	0	0	0	0.3	0.3	0	0	0	0	0.3	0.2	0.74	EN
Linaria flava (Poir.) Desf. subsp. *sardoa* (Sommier) A.Terracc.	0	0	0	0.03	0.3	0.2	0	0	0	0	0.5	0.4	0.70	EN
Euphrasia marchesettii Wettst. ex Marches.	0	0	0	0	0.5	0.4	0	0	0	0	0.5	0.4	0.56	EN
Carex panormitana Guss.	0	0	0	0	0.5	0.5	0	0	0	0	0.6	0.5	0.53	EN
Saxifraga tombeanensis Boiss. ex Engl.	0	0	0	0	0.8	0.8	0	0	0	0	0.8	0.8	0.24	EN
Pilularia minuta Durieu ex A.Braun	0	0	0	0	0.4	0.3	0	0	0	0	0.4	0.3	0.68	VU
Campanula sabatia De Not.	0	0	0	0	0.8	0.7	0	0	0	0	0.8	0.7	0.28	VU
Saxifraga florulenta Moretti	0	0	0	0	0.9	0.8	0	0	0	0	0.8	0.8	0.19	VU
Crocus etruscus Parl.	0	0	0	0	0.3	0.3	0	0.01	0	0.16	0.2	0.2	0.79	NT
Fritillaria montana Hoppe ex W.D.J.Koch	0	0	0	0	0.6	0.5	0	0	0	0	0.6	0.5	0.48	NT
Brassica glabrescens Poldini	0	0	0	0	0.6	0.5	0	0	0	0	0.6	0.6	0.43	NT
Adenophora liliifolia (L.) A.DC.	0	0	0	0	0.7	0.6	0	0	0	0	0.7	0.6	0.40	NT
Gladiolus palustris Gaudin	0	0	0	0	0.6	0.6	0	0	0	0	0.7	0.6	0.39	NT
Gentiana lutea L. subsp. *lutea*	0	0	0	0	0.7	0.6	0	0	0	0	0.8	0.7	0.31	NT
Saxifraga presolanensis Engl.	0	0	0	0	0.8	0.8	0	0	0	0	0.8	0.7	0.25	NT
Brassica insularis Moris	0	0	0	0	0.9	0.9	0	0	0	0	0.9	0.9	0.12	NT
Stipa austroitalica Martinovský	0	0	0	0.01	0.3	0.2	0	0.03	0	0.39	0.2	0.1	0.84	LC
Campanula morettiana Rchb.	0	0.002	0.002	0.1	0.3	0.2	0	0	0	0	0.4	0.3	0.76	LC
Selaginella denticulata (L.) Spring	0	0	0	0	0.3	0.2	0	0	0	0	0.4	0.3	0.74	LC
Gentiana ligustica R.Vilm. & Chopinet	0	0	0	0	0.4	0.3	0	0	0	0	0.4	0.3	0.65	LC
Ophrys lunulata Parl.	0	0	0	0	0.4	0.4	0	0	0	0	0.4	0.3	0.65	LC
Daphne petraea Leyb.	0	0	0	0	0.5	0.4	0	0	0	0	0.5	0.4	0.58	LC
Selaginella selaginoides (L.) P.Beauv. ex Schrank & Mart.	0	0	0	0	0.6	0.5	0	0	0	0	0.4	0.3	0.58	LC
Gentiana lutea L. subsp. *vardjanii* Wraber	0	0	0	0	0.5	0.5	0	0	0	0	0.5	0.4	0.55	LC
Artemisia genipi Weber ex Stechm.	0	0	0	0	0.6	0.5	0	0	0	0	0.5	0.5	0.51	LC
Cypripedium calceolus L.	0	0	0	0	0.6	0.6	0	0	0	0	0.5	0.4	0.51	LC
Lycopodium clavatum L.	0	0	0	0	0.5	0.5	0	0	0	0	0.6	0.5	0.51	LC
Selaginella helvetica (L.) Spring	0	0	0	0	0.5	0.4	0	0	0	0	0.6	0.6	0.49	LC
Arnica montana L. subsp. *montana*	0	0	0	0	0.7	0.6	0	0	0	0	0.6	0.5	0.47	LC
Huperzia selago (L.) Schrank & Mart.	0	0	0	0	0.6	0.6	0	0	0	0	0.7	0.6	0.41	LC
Aquilegia alpina L.	0	0	0	0	0.8	0.7	0	0	0	0	0.5	0.5	0.40	LC
Asplenium adulterinum Milde subsp. *adulterinum*	0	0	0	0	0.6	0.6	0	0	0	0	0.7	0.7	0.37	LC
Lycopodium annotinum L. subsp. *annotinum*	0	0	0	0	0.7	0.6	0	0	0	0	0.7	0.6	0.37	LC
Physoplexis comosa (L.) Schur	0	0	0	0	0.7	0.6	0	0	0	0	0.7	0.6	0.37	LC
Dianthus rupicola Biv. subsp. *rupicola*	0	0	0	0	0.7	0.7	0	0	0	0	0.8	0.7	0.29	LC
Primula polliniana Moretti (= *P. spectabilis* Tratt.)	0	0	0	0	0.8	0.7	0	0	0	0	0.8	0.7	0.29	LC
Diphasiastrum alpinum (L.) Holub	0	0	0	0	0.6	0.5	0	0	0	0	0.5	0.5	0.52	DD

From a conservation management perspective, this study suggests that the considered Italian policy species are less vulnerable, at least to climate change, than though in the past. Hence, conservation efforts should be directed mainly toward reducing the impact of fragmentation and other human-induced threats as those connected to agriculture and land-use change (Thuiller, 2007). The reinforcement of current ecological corridors may naturally reduce the risk of extinction for widespread species with boleochory dispersal capacity. A limitation of our study is that the model included the current land use only, but drastic landscape changes will likely occur in the next 50 years. Settlement and infrastructure development are likely to further reduce the occurrence of natural habitats along the coasts, and forests are predicted to further expand in mountain areas (Borana & Yadav, 2017; Falcucci et al., 2007; van Gils, Orgil, Rossiter, Wizaso, & Liberatoscioli, 2008). Therefore, the predicted range loss found in this study may be underestimated for some of the target *taxa*. Moreover, in interpreting our results, we must consider the possible bias due to at least four further limitations that generally affect SDMs. The first is represented by the difficulties in modeling the contribution of competition and facilitation in the colonisation of new areas and range shifts. Indeed, species requirements in terms of biotic interactions may prevent their establishment in places that are suitable from an abiotic viewpoint (Liang, Duveneck, Gustafson, SerraDiaz, & Thompson, 2018). Analogously, our model does not parametrize some relevant components of the species establishment niche. For instance, we do not consider whether the requirements for seed germination and seedling establishment for the target species are met in the potential colonisation areas. Indeed, seed germination and seedling establishment is a key phase of plant development, and it is strongly affected by climate change both positively (Mondoni et al., 2015) or negatively (Marcante, Sierra-Almeida, Spindelböck, Erschbamer, & Neuner, 2012). The scale of our analyses can represent a further source of bias. Our model is based on 1 km^2 climatic data, but plants growing in highly heterogeneous habitats (e.g. cliffs, mountain slopes, etc.), may require finer modeling scales. Accordingly, in spite of climate change, fine-grain distribution models suggest improved chance of long-term persistence of cold-adapted species in high-elevation sites (Randin et al., 2009). Finally, we must take in account the possible effect of gaps in occurrence data, because this issue can variously affect the accuracy of SDMs. Nonetheless, in our case occurrence data were very precise (all occurrence points were collected during ad-hoc monitoring programs) and recent; therefore, we can expect a minor effect on our results. Some uncertainties are also related to the climate change scenario that will occur in the near future. Moreover, defections from the Paris Agreement on climate change (under the United Nations Framework Convention on Climate Change 2017) may favor the worst scenario.

The shift of species distribution ranges due to climate change will pose interesting challenges to the Natura 2000 network of protected areas, which is now very effective in protecting policy species and other threatened *taxa*, but lacks the flexibility to cope with species range shift

(Araùjo et al., 2011; Fois et al., 2018). It is therefore important to assess whether current conservation and management strategies (e.g. action plans) will still be effective in the future. In other cases (e.g., mountain plants), more extreme conservation actions like assisted colonization may also be considered (Haskins & Keel, 2012). However, assisted colonization is still a quite controversial measure for species highly threatened by climate change (McLachlan, Hellmann, & Schwartz, 2007; Ricciardi & Simberloff, 2009), and it can be unsuccessful for long-term conservation, as demonstrated for marginal populations of *Carex foetida* in the northern Apennines (Ferrarini et al., 2016).

In conclusion, stochastic SDMs incorporating data on plant dispersal, generation length, and habitat fragmentation can constitute a significant improvement in predicting range shift due to climate change (Miller et al., 2015). Lack of data and incomplete knowledge prevent the coupling of SDMs with demographic models in the application of RL criteria to assess extinction risk due to climate change (Stanton et al., 2015). In such cases, our approach appears to be most feasible, in which the RL assessment is integrated with the outcomes of the predictions of range shift to prioritize and further define appropriate conservation actions. Considering the position of Italy in the center of the Mediterranean biodiversity hotspot, the situation depicted in the present study may be representative of the policy species of other EU Mediterranean countries as well as of policy species occurring in the southern European mountain chains (considering that most of the assessed taxa are mountain species). Similar analyses based on national red lists in these countries (Jelić et al., 2012; Moreno Saiz et al., 2003) will be very useful to broaden the framework of conservation planning in this biodiversity hotspot.

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